10.2 CHEMICAL-SPECIFIC CONCEPTUAL SITE MODELS

The CSMs for the 13 indicator contaminants are presented in the following sections. Each discussion includes a brief presentation of contaminant distribution, potential sources and pathways, and loading, fate, and transport. For each CSM, a three-section panel series is provided that presents cross-media contaminant distributions and available source information and presents a subset of the contaminant distribution data. An electronic version of the three-section CSM panels is included in Appendix I.

Panel A presents summary information on the observed concentration in surface sediment using Thiessen polygons to spatially represent concentrations between data points, sediment traps, riparian soil/sediment, surface water), and TZW from RM 1.9 to 11.8, excluding dredge and cap sample locations. A histogram of the surface sediment data is included in the top left-hand corner of each panel to show the distribution of the data. Unfiltered push probe, filtered push probe, and peeper results are displayed for TZW. Surface water XAD data are presented for total PCBs, dioxins/furans, total DDx, total PAHs, total chlordanes, aldrin, and dieldrin. Surface water peristaltic pump data are presented for arsenic, chromium, copper, zinc, and TBT. The BEHP data presented are a combination of the XAD and peristaltic pump data.

Panel B presents a summary of subsurface sediment concentrations and large-scale (>30 cm) erosional/depositional areas predicted for a major flood based on the FS HST model for the Study Area (Chen 2011, pers. comm.). Thiessen polygons on these panels represent concentrations in the sediment interval just below the surface sediment (typically the B interval). A histogram of the subsurface sediment data is included in the top left-hand corner of each panel. Also included are icons depicting the locations of 10 major types of historical industries that are or were active in the Study Area.

In addition to the erosional/depositional information shown on Panel B, Map 10.2-1 shows areas in the Study Area at risk for surface sediment disturbance from incidental anthropogenic activities based on water depth and in-water operations. This includes all areas above the –5 ft NAVD88 contour that are potentially subject to boat wakes, areas in the immediate vicinity of docks and berths, and any additional areas where sediment scour that did not appear to be due to natural forces was evident in the 2002–2009 bathymetric time-series data set.[1] This map does not include an analysis of those activities that are specifically intended to move sediments (shoreline/structure construction, maintenance dredging, or remedial capping/dredging).

Panel C presents whole-body concentrations in field-collected smallmouth bass, clams, crayfish, and sculpin. A detailed view of the composite groupings can be found on Map 2.2-10a–d.

[1] Map 10.2-1 is a qualitative presentation of areas where there is a reason to believe that anthropogenic disturbance risk may be relatively higher than other areas.

For each upland site that has undergone sufficient investigation to identify known or likely complete pathways, a box is shown on the panels listing the applicable pathways and noting whether they are complete or likely complete. Where there are insufficient data to make a determination or when a complete pathway was determined to be not present (see Tables 10.2-1 and 10.2-3 through 10.2-14), no information is presented.

Along with the panels, three figures are provided for each CSM chemical to portray loading, fate, and transport processes under current conditions in the Study Area[2]. The first figure consists of a pair of box and whisker plots; the first compares the range of the estimated external and internal annual loads to the Study Area for each of the loading terms quantified for a given CSM contaminant, while the second compares the concentrations of the chemical in surface sediment, sediment traps, and suspended solids in surface water for the entire Study Area. The second figure is a box-and-arrow diagram[3] depicting relevant loading, fate, and transport processes for each CSM chemical at the Study Area scale. The third figure provides a graphical comparison, by river mile, of the quantified external and internal loading terms, including central, upper, and lower estimates, and affords additional resolution of spatial patterns in loading to the Study Area.

As documented in Section 6.1 and Appendix E, external loading of each CSM contaminant to the Study Area was estimated quantitatively for upstream surface water,[4] stormwater, atmospheric deposition to the water surface, and groundwater advection through subsurface sediments. Quantitative estimates were also generated for upland groundwater plumes and permitted point-source discharges for a subset of the CSM contaminants for which these terms may be significant. Unquantified loading terms, including bedload, volatilization, and riverbank erosion, are represented qualitatively on the box-and-arrow diagrams. The only contaminant fate and transport mechanism internal to the Study Area for which quantitative estimates were developed in the RI is pore water advection from surface sediment to the overlying surface water column. Other internal fate and transport mechanisms, including sediment erosion, sediment deposition, sediment burial, and biological and geochemical transformation (degradation) are represented qualitatively on the box-and-arrow diagrams.

10.2.1 Total PCBs

The Study Area graphical CSM for PCBs is presented on Panels 10.2-1A–C. PCBs are a class of nonpolar, synthetic, halogenated hydrocarbons that were manufactured in the

[2] Includes surface water and bedded sediments in the surface mixed layer (0–40 cm bml).

[3] This diagram does not attempt a mass balance because sufficient data are not available and because of the varying levels of quantification (qualitative to quantitative) of each term.

[4] As discussed in Section 6.1, estimated upstream surface water loads were developed using data from both RM 11 and RM 16. Because of the complex hydrodynamics on the LWR between its confluence with the Columbia River (RM 0) and the entrance to Multnomah Channel at RM 3 (frequent flow reversals, see Section 3.1.3.3), surface water chemical loads leaving the Study Area at RM 1.9 could not be estimated using the simplified approach described in Section 6.1. The furthest downstream surface water loads for the LWR were estimated at RM 4. Surface water loads exiting the Study Area via Multnomah Channel were also estimated.

United States between 1929 and 1977 and widely used for a variety of purposes. Current allowed uses include transformers, heat transfer systems, natural gas pipelines, existing carbonless copy paper, and electrical switches. Historical PCB uses included dielectric fluids in transformers and capacitors, electrical cables, cutting oils, hydraulic oils, lubricants, heat transfer fluids, plasticizers, flame retardants, additives to pesticides, paints, carbonless copy paper, caulk, adhesives, sealants, in heat transfer systems, electromagnets, and for dust suppression. PCB sources in waste materials include scrap metal recycling, auto salvage, used oil, recycled paper, asphalt roofing materials, building demolition, and in the repair and salvaging of ships, locomotives, heavy equipment, and manufacturing equipment. Although PCBs are ubiquitous in the environment, commercial PCB production in the United States ended in 1977. Consequently, most of the mass of PCBs found in the Study Area sediments is primarily derived from historical sources. In addition, secondary sources may introduce PCBs to the lower Willamette River through a variety of environmental pathways as described in Section 10.2.1.2

Although PCBs do degrade in the environment (e.g., by reductive dehalogenation), they are persistent. PCBs are hydrophobic/lipophilic organic substances that accumulate in organisms both by uptake from the environment over time (bioaccumulation) and along the food chain (Erickson 1997). PCBs biomagnify with each trophic level in the food web (biomagnification). In aquatic organisms, the rate and physiological mechanism of PCB metabolism depend on the species and the specific type of PCB.

~~As PCBs are no longer produced and sold, current significant releases to the environment are uncommon; however, relatively small PCB releases still occur. Consequently, most of the mass of PCBs found in the Study Area sediments is primarily derived from historical sources. In addition, secondary sources may introduce PCBs to the lower Willamette River through a variety of environmental pathways as described in Section 10.2.1.2.~~

Commented [KK1]: This was moved to end of paragraph above.

~~10.2.1.2~~10.2.1.1 PCB Contaminant Distribution

On a harbor-wide basis, the highest PCB sediment concentrations occur in nearshore areas and in locations proximal to local upland sources (Map 5.1-2a–m and Panels 10.2-1A–B). Relatively high concentrations of PCBs are also often found in riparian sediments, sediment trap samples, surface waters, and biota samples in the areas with elevated sediment concentrations.

Similar spatial and concentration trends are observed in subsurface sediments (Panel 10.2-1B). Areas where surface and subsurface concentrations are not well correlated may be an indication of spatially and temporally variable inputs and sources, or to different influences from sediment transport mechanisms. Areas where the highest concentrations of PCBs in sediment are observed include RM 11.3E, RM 8.8-10W, Swan Island Lagoon, International Slip (RM 3.7-3.8E), RM 2.1-2.5E, and RM 4.0 to 4.1E. Total PCB concentrations are generally higher in subsurface sediments (Panels 10.2-1A–B, Maps 5.1-2a–m, and Figure 5.1-33), pointing to predominantly historical total PCB sources and higher past loads. but exceptions to this trend are noted at RM 11E, Swan Island Lagoon, and Willamette Cove.

Exceptions to the pattern of higher concentrations at depth are found at RM 11.3E, in Willamette Cove, and in Swan Island Lagoon. PCB concentrations in area of RM 11.3 are greater in surface than subsurface levels along the eastern nearshore area and adjacent channel edge. The nearshore sediment PCB distribution, as well as the collocated surface water and sediment trap data, indicate a local, recent input and/or redistribution of PCBs historically released into this area and present in the sediments. Anthropogenic sediment disturbance in this area also may have altered the distribution of PCBs with depth in the sediment column and re-exposed and re-mobilized subsurface contamination.

Collocated surface and subsurface samples from the inner portion of Willamette Cove also exhibit higher surface PCB concentrations than at depth which may be indicative of higher recent inputs. Finally, in Swan Island Lagoon mean surface and subsurface total PCBs concentrations are approximately the same. The lack of a vertical gradient may reflect a combination of time-varying inputs, low net sedimentation rates, and localized high surface sediment mixing rates that result in variable spatial trends in sediment quality with depth.

Relatively higher concentrations of PCBs in surface water are generally found in areas with elevated sediment concentrations. The highest total PCB concentrations were associated with single point samples in Willamette Cove and at RM 5.5. Total PCB concentrations at RM 11 were consistently higher than at RM 16, suggesting a source or sources between these locations. Aside from these exceptions, the range of total PCB concentrations within the Study Area surface water was fairly consistent.

Total PCB congener concentrations in the Study Area sediment trap samples were ~~all~~ one-to-five fold greater than upstream concentrations~~, in most cases~~. These trends were ~~generally~~ also reflected in the Aroclor data. ~~The d~~Downstream total PCB concentrations are ~~generally the same as the~~ similar to concentrations observed in Multnomah Channel and ~~seem to be approaching~~ the upriver reach. ~~concentrations, although the total PCB congener concentrations are about two fold higher.~~ The concentrations entering the site, at least in the eastern nearshore region are similar in concentration to the upriver sediment traps ~~indicating that~~ for the time periods measured.~~, the downtown reach has little to no influence on the incoming depositional sediment concentrations.~~

High concentrations of PCBs were typically found in biota samples in areas with high sediment concentrations.

Commented [KK2]: To be consistent with discussions of other media, there should be a discussion of which biota samples and where to corroborate this statement.

Spatial variations in PCB composition (based on congener data) are evident throughout the Study Area, and areas of elevated PCB sediment concentrations often exhibit congener homolog patterns that are distinct from surrounding areas of lower PCB concentrations (Maps 5.1-32 and 5.1-33; Figures 5.1-35a–c and 5.1-36a–c). PCB homolog patterns in surface and subsurface sediment, sediment traps, and in the particulate portion of the surface water samples, are often similar within each area. Subsurface sediment patterns are less consistent with surface sediment homolog patterns for the areas at RM 6.9 to 7.5W and RM 2.1 to 2.5E when compared other contaminated locations.

Relatively low concentrations of PCBs are widespread in portions of the harbor away from the localized areas of elevated concentrations (Panel 10.2-1A and Map 5.1-1). The homolog patterns in these widespread, low level PCB areas are generally similar in both depositional and erosional areas, have less distinct variations than areas of higher concentration, and may reflect inputs from upstream and transport within the Study Area.

~~PCB concentrations in sediments across much of the Study Area range from nondetected to about 34 μg/kg. This is about # times the upriver background level of XX μg/kg corrected for the organic carbon content.~~

Commented [Int3]: Value to be added following dispute resolution. EPA CSM guidance indicates background levels should be noted in conjunction with site information.

~~10.2.1.4~~10.2.1.2 Potential PCB Sources and Pathways

Numerous upland sites have been identified as being known or likely historical and/or current sources of PCBs. These sites discharge directly to the river or discharge through shared conveyance systems (Table 4.4-1). Historical and current known or likely complete pathways for PCBs in stormwater have been identified at several properties associated with former shipyards and sites where transformers were serviced and/or dismantled. Historical wastewater discharges associated with ship building and decommissioning, electrical component manufacturing, and leaks and spills from equipment that used fluids that may have contained PCBs are likely, but have not been specifically identified or quantified. Wastewater discharges are currently regulated primarily through NPDES permits.

Potential upland and overwater sources and identified known and likely complete migration pathways are identified on Table 10.2-1 and Panels 10.2-1A–C. These sources and pathways, identified on the basis of the process described in Section 4.2, focus on ECSI sites and are based on a review of information in the associated DEQ ECSI files and other readily available site information, including, in the case of LWG-member sites, information provided by the site owner.

The most significant migration pathways for PCBs in the Study Area are historical and included industrial wastewater, stormwater, overland transport, overwater releases, and riverbank erosion (Table 10.2-1). Atmospheric deposition and upstream inputs may have also contributed PCBs to the Study Area, the releases from these pathways are not quantifiable and are difficult to distinguish.

Stormwater PCB loads have decreased substantially from historical levels since implementation of stormwater controls and the statutory ban on PCB manufacture in the 1970s. Overland transport was likely more important prior to the development of extensive stormwater conveyance systems. Bank erosion is also likely more important when PCBs were in wider use, or when contaminated material was used in construction fill activities. Historical PCB overwater releases have not been identified through the file review process, but are likely to have occurred in association with overwater operations, such as ship building and dismantling, ship repair and maintenance, and with the use of hydraulic fluids in dock operations. The locations of elevated PCB concentrations in sediments coincide in some cases with ship construction, dismantling, and repair operations, and it is likely that overwater releases occurred concurrently as a result of historical activities in these locations. PCBs are also detected in sediments near outfalls draining facilities historically engaged in electrical equipment manufacturing, such as at RM 11.3E and at OF-17. Stormwater discharges and riverbank erosion associated with fill soil from offsite and/or steel manufacturing activities at RM 2E have also resulted in PCB contamination in sediments.

Current PCB inputs to the Study Area are lower than historical inputs. However, measured elevated levels of PCB concentrations in surface sediments and other media, including biota in the International Slip and Swan Island Lagoon indicate ongoing localized inputs and/or internal mass transfer of historical PCB inventory from subsurface to surface sediments and then to other media by processes such as sediment resuspension (due to both natural and anthropogenic disturbance factors) and biological uptake. While surface sediments generally exhibit lower PCB concentrations than subsurface sediments, the temporal persistence of elevated PCB levels in surface sediments in many nearshore and off-channel areas suggests that net sedimentation rates may be low in many nearshore areas. This is supported by the bathymetric change data and the limited radioisotope data from the Study Area (Anchor 2005), sediment column mixing rates are high, and inputs of PCBs still occur. Potentially important current pathways include stormwater and riverbank erosion. The effects of current releases and these physical site features are expressed in surface sediment concentrations (Panel 10.2-1A).

Known or likely complete historical pathways for PCBs have been identified at 40 sites (Table 10.2-1 and Panels 10.2-1A–C), and include stormwater (38 sites), overland transport and riverbank erosion (6 sites), and overwater releases (1 site).

Current known or likely complete pathways for PCBs have been identified at 18 sites, and include stormwater (16 sites), overland transport (2 sites), riverbank erosion (3 sites,) and groundwater transport (1 site).

~~10.2.1.5~~10.2.1.3 Loading, Fate, and Transport of PCBs

PCB loading, fate, and transport in a typical year for the Study Area is summarized on Figures 10.2-1a-b, 10.2-2, and 10.2-3. Estimates are for current conditions and likely differed historically. Much of the PCB mass in the Study Area, especially in deeper sediments, is attributable to historical loading that occurred under different loading conditions and rates.

Upstream surface water represents the largest estimated current loading term for PCBs to the Study Area (Figures 10.2-1a and 10.2-2), and is associated with both the dissolved and suspended fractions. On a annually averaged basis, the majority of this load occurs during low-flow conditions (Figure 6.1-2) which exist for approximately two-thirds of the year. Surface water samples collected during high-flow consistently exhibited lower concentrations of PCBs than in low-flow samples, indicating that inflow concentrations at high flow rates have greater influence than local effects. Total PCB concentrations in sediments accumulating in upstream borrow pits, which are likely a mixture of upstream bedload and suspended load, are comparable to the upriver bedded sediment background concentrations.

PCB loads in surface water increase between the upstream and downstream boundaries of the Study Area in both the particulate and dissolved fractions, and is in part attributable to quantified external loads (stormwater, atmospheric deposition, and advection through subsurface sediments). Other possible reasons for the increased loading is due to internal fate and transport processes such as sediment resuspension, which have not been quantified. The distribution of total PCBs in surface sediments, sediment trap samples, and the particulate fraction of surface water samples, on both a dry-weight and OC-normalized basis, is presented on Figure 10.2-1b.

Estimates of current PCB loading via stormwater are approximately half the estimated upstream load, atmospheric deposition directly to the Study Area river surface (is nearly an order of magnitude lower than the upstream surface water load (Figure 10.2-2). PCBs were detected in stormwater in Round 3A and 3B sampling in each land use area sampled (see Section 6.1.2.3 and 10.1.4.1.2). Groundwater advection through subsurface sediments is estimated the least significant of the quantified terms, but is subject to a relatively high degree of uncertainty due to the variability in published organic-carbon partitioning values for PCBs. As discussed in Section 6.1.1.2, bedload into- and out of the Study Area is expected to be low relative to dissolved and particulate surface water loading. PCB volatilization from the water column is relevant for only a small fraction of the less chlorinated PCB congeners and is also expected to be low.

Fate and transport processes internal to the Study Area for total PCBs are shown on Figure 10.2-2. PCB transport to the water column due to pore water advection through surface sediments is only process for which quantitative estimates were developed, and is estimated to be similar in magnitude to the subsurface advective loading. Other internal fate and transport processes are depicted on Figure 10.2-2 on a qualitative basis only. Sediment erosion, deposition, and burial are a function of locally and temporally variable hydrodynamic conditions and the surface sediment mixing rate. PCB partitioning between suspended sediment and surface water depends on the relative concentrations associated with suspended particulate organic carbon and the dissolved surface water fraction, as well as reaction kinetics. The fate of PCBs within the Study Area may also be influenced by relatively slow microbially-facilitated degradation and photolysis.

PCB loads from upland groundwater plumes are not expected to be significant and estimates were not generated. Loading from permitted point source discharges were not estimated because PCBs are not regulated and monitored under any active discharge permits within the Study Area. Estimates of PCB loading from upland soil and riverbank erosion also were not assessed due to a paucity of data for riparian soil PCB concentrations and erosion rates.

The total PCB load in surface water increases downstream through the Study Area to RM 4[5] (Figure 10.2-3), the largest PCB stormwater inputs enter the Study Area between RM 3 and 4. As described in Section 6.1.2, the estimated load (1 kg/yr) to this reach is largely from a non-representative (unique) site (Outfall WR-384) and exceeds the next highest stormwater load, between RM 8 and 9, by a factor of 10.

Atmospheric deposition is estimated to contribute a total PCB load approximately one-third that of stormwater at the Study Area scale. Deposition loading to the water surface varies only as a function of water surface area by river mile.

No current known or likely complete PCB overwater pathways have been identified. Current overwater releases may be locally important at sites with continuous waste handling or operational activities, but are considered a minor current pathway overall.

While areas of PCB contamination in the Study Area appear to be the result of releases from specific localized sources, PCBs also enter the Study Area from non-point and diffuse sources such as private and public stormwater and sewer outfalls, and sources upstream of the Study Area in the Willamette River. Harbor-wide loading estimates indicate that the highest current external inputs to the Study Area are from upstream surface water and, to a lesser degree, stormwater runoff and atmospheric deposition. These estimates indicate that the mass flux of PCBs in surface water exiting the Study Area at RM 2 and at the Multnomah Channel entrance exceed the estimated fluxes entering the Study Area from all quantified external loading terms. This indicates an internal mass transfer of PCBs from bedded sediment to the water column, likely through sediment resuspension.

~~10.2.1.6~~10.2.1.4 Human and Ecological Risks Associated with PCBs

PCBs are a primary contributor to human health risks at Portland Harbor, primarily through consumption of resident fish, and to breast-fed infants whose mothers have been exposed to PCBs. Total PCBs pose ecologically significant risks to mink, river otter, and spotted sandpiper, and low risk to osprey, bald eagle, sculpin, and ~~smallmouth bass~~multiple fish species. Total PCBs pose risks to tecological receptors through more liens of evidence than did any other contamined evaluated in the BERA. They were also identified as one of the primary contaminants in sediment ~~likely to pose~~ posing ~~potentially~~ unacceptable risk to the benthic community.

[5] At approximately RM 3, the Columbia River and Multnomah Channel hydraulically influence the flow regime complicating interpretation of load conditions in this area (see Section 3.1.3.3).

10.2.2 Total PCDD/Fs

A graphical CSM for Total PCDD/Fs in the Study Area is presented Panels 10.2-2A–C. As a group, PCDDs represent 75 different isomers, while PCDFs comprise over 135 compounds (Eisler 1986). These two chemical classes are generally referred to as dioxins and furans, respectively. PCDD/Fs are byproducts of combustion, incineration, certain industrial chemistry processes, and natural sources, including combustion, metal smelting, and production bleached paper, polyvinyl chloride (PVC), inks/dyes, certain chlorine production technologies, chlorophenols, chlorinated herbicides, and commercial Aroclor (PCB) mixtures (ATSDR 1998). Examples of combustion and incineration that may lead to the formation of PCDD/Fs include hazardous or medical waste incinerators, cement kilns, boilers and industrial furnaces, vehicle emissions, fossil fuel power plants, burning PCB containing electrical equipment (such as transformers), and backyard burning (e.g., refuse piles, burn barrels). PCDD/Fs are naturally produced from forest fires, volcanic eruptions, and sedimentary deposits. Currently the largest source is from backyard burning (EPA 2006b). When released into the air, some PCDD/Fs may be transported long distances, even around the globe. In the atmosphere, it has been estimated that 20 to 60 percent of 2,3,7,8-TCDD is in the vapor phase. Sunlight and atmospheric chemicals can break down a very small portion of the PCDD/Fs, but most PCDD/Fs will be deposited on land or water (ATSDR 1998).

Behavior of PCDD/Fs in the environment is characterized by low vapor pressures, high octanol-water and organic carbon partitioning coefficients (K_{ow} and K_{oc}, respectively), and extremely low water solubilities. They are hydrophobic and have a strong affinity for sediments with high organic matter content; as a result, transport of PCDD/Fs in aquatic systems is closely tied to fine-grained sediment transport processes. Some PCDD/Fs, present near the water surface and not bound to solids, may be subject to photodegradation (EPA 1994). In general, PCDD/Fs are stable in all environmental media, with persistence measured in decades. Chemical degradation of PCDD/Fs through reductive dechlorination can also occur, but it is typically a slow process. PCDD/Fs have been shown to biomagnify in aquatic food webs and associated avian and mammalian species (ATSDR 1998).

10.2.2.1 Total PCDD/F Contaminant Distribution

Panels 10.2-2A–C present the Study Area graphical CSM for Total PCDD/F. Care should be taken in interpreting Panels 10.2-2A–B because relatively few sediment samples were analyzed for PCDD/Fs. Total PCDD/Fs were detected in sediments in several locations along the eastern and western nearshore zones and in Swan Island Lagoon. The highest detected concentrations were found in the eastern nearshore zone at RM 2E-8E, Swan Island Lagoon, RM 11E, RM 6W-10.3W, from RM 4W-6W, and at RM 3.4W. There are a number of distinct locations scattered throughout the Study Area that exhibit elevated PCDD/F concentrations in sediment and coincide with currently identified known or likely historical industrial dioxin and/or furan sources, including RM 11E, Swan Island Lagoon, Willamette Cove, and between RM 6.5W and 7.5W, adjacent to Arkema and Rhone-Poulanc. Total PCDD/F concentrations in the subsurface are generally greater than that observed in surface sediments. The higher concentrations observed in subsurface sediment relative to concentrations in surface sediment are indicative of a primarily historical input of these contaminants to the Study Area. Areas of apparent PCDD/F contamination in sediment in other locations in the Study Area not associated with documented sources and pathways that all point sources may not have been identified. However these locations are coincident with areas exhibiting ~~high~~ ~~elevated~~higher concentrations of other indicator contaminants, such as PCBs and/or PAHs. While the relatively low density of PCDD/F data makes surface to subsurface concentration gradients difficult to discern on a Study Area-wide basis, there does not appear to be strong trends with depth.

~~Detected PCDD/F concentrations in most Study Area surface sediments range from 0 to 500 pg/g. This compares to upriver background levels of XX pg/g (OC-equivalent basis).~~

Commented [Int4]: Value to be added following dispute resolution

Surface water concentration are elevated in Willamette Cove, at RM 6.8W, and in the RM 4 transect sample. Total PCDD/F surface water concentrations within the Study Area did not display any consistent trends from upstream to downstream.

There is no strong spatial or temporal gradients evident in total PCDD/F concentrations measured in suspended sediments collected in sediment traps within the Study Area (Figure 5.2-14). In general, Study Area sediment trap samples had higher total PCDD/F concentrations than the upstream locations near RM 15.6.

PCDD/Fs were detected in all fish and invertebrate tissue samples collected from the Study Area, the highest concentrations were observed in samples collected between RM 6.5 and 7.5 (Panel 10.2-2C).

10.2.2.2 Potential PCDD/Fs Sources and Pathways

Historical known complete or likely complete pathways for PCDD/Fs have been identified at four sites (Table 10.2-3 and Panels 10.2-2A–C) and include stormwater (four sites), groundwater (one site), overwater releases (one site), overland transport (two sites), and riverbank erosion (two sites).

Current known complete or likely complete pathways for PCDD/F have been identified at two sites and include stormwater (both sites), overland transport (one site), and riverbank erosion (one site). Stormwater transport is expected to be the most significant current pathway for PCDD/Fs to enter the Study Area from adjacent upland sites. A current known complete pathway has been identified for the City of Portland Outfall OF-22B~~ and Triangle Park property~~. Gould Electronics, Rhone Poulenc, and McCormick & Baxter are identified as having known complete historical pathways.

Commented [Int5]: What is the basis for EPA strike out of Triangle Park here?

Commented [KK6R5]: Triangle park is not a significant source of PDDD/F.

No current known or likely PCDD/F overwater pathways have been identified. However, McCormick & Baxter is identified as having a known historical pathway.

No sites have been identified as having current known or likely complete groundwater pathways for PCDD/Fs, though McCormick & Baxter had a historical complete pathway, the potential for migration in the past is known to have existed at McCormick & Baxter.

10.2.2.3 Loading, Fate, and Transport of PCDD/Fs

The loading, fate, and transport assessment for total PCDD/Fs is summarized on Figures 10.2-4a, 10.2-5a, and 10.2-6a. The greater PCDD/Fs mass in the Study Area deeper sediments (Panel 10.2-2B) is likely attributable to historical loading that occurred under different past loading conditions and rates. Due to the limited data available, PCDD/F loading terms were quantified only for upstream surface water and advection through subsurface sediments. Upstream loading contributes much greater estimated loads of total PCDD/Fs to the Study Area than advection through subsurface sediments (Figures 10.2-4a and 10.2-5a). Upstream surface water loads are associated primarily with the suspended particulate fraction, with similar total PCDD/F mass loads entering the Study Area during high-flow and low-flow conditions (see Figure 6.1-4). The total PCDD/F upstream loads in surface water comparable with downstream loads. A cross-media comparison and statistical assessment of surface sediment, sediment trap samples, and the particulate fraction of surface water samples on a Study Area-wide basis is shown in Tables 10.2-2a-b and Figure 10.2-4b. An apparent increase in the estimated surface water flux of PCDD/Fs from upstream to downstream within the Study Area may reflect contributions from a combination of other external loading terms (stormwater, advective transport, and atmospheric deposition) or internal fate and transport processes, such as sediment resuspension.

10.2.2.4 Human and Ecological Risks Associated with PCDD/Fs

Dioxins and furans pose unacceptable risks to human health at Portland Harbor, primarily due to exposures through consumption of shellfish and resident fish, as well as a direct contact exposure to contaminanted sediments for peole engaged in fishing activities. Dioxins and furans poses an ecologically significant risk to mink, ~~and risk of adverse effects to river otter,~~ spotted sandpiper, ~~and~~ osprey, ~~to~~ and bald eagles. They are also contributors to the overall risk to river otters.

10.2.2.5 DDx

The graphical CSM for DDx compounds in presented in Panels 10.2-3A–C,loading information is summarized in Figures 10.2-7 through 10.2-9.

DDT was widely used as an insecticide from about 1943 (Porter 1962) until 1972, when it was banned for most uses in the United States, because of its toxicity to wildlife-. DDE and DDD are the primary metabolites of DDT, but technical-grade DDT may also contain DDE and DDD as impurities (ATSDR 2002d). DDT was released historically to air and soil through widespread spraying of crops and forests, and for mosquito control. Releases also occurred at more local scales at pesticide manufacturing and storage facilities. Ongoing releases occur in countries where its use is not banned, and some of these releases can be transported globally through the atmosphere. Because

DDT is no longer produced or sold in the United States, significant new releases to the environment are uncommon. Consequently, most of the mass of DDx found in the Study Area is derived from historical sources. However, DDx compounds continue to be introduced to the LWR through a variety of environmental pathways, as described in Section 10.2.3.2.

DDT degrades slowly via abiotic and microbially-mediated processes to the more persistent DDx compounds, DDE (under aerobic conditions) and DDD (in anoxic systems; EPA 2000b). In the environment, DDx compounds are persistent and are readily bioaccumulated in aquatic organisms (EPA 2000b). While there is no clear evidence that DDT, DDD, or DDE causes cancer in humans, there is sufficient evidence of carcinogenicity of these substances in rodents, which has led to their classification as probable human carcinogens (ATSDR 2002d).

10.2.2.6 DDx Contaminant Distribution

Within the Study Area, the highest DDx concentrations in sediments are limited to localized areas in nearshore zones. The highest reported concentrations in sediment are located in the western nearshore zone between RMs 6W and 7.5W, and are proximal to known upland sources. Other areas of elevated total DDx sediment concentrations are smaller in extent and are located at RM 8.8W, at the mouth of Swan Island Lagoon, and in subsurface sediments only at RM 4.8W and the head of the International Slip.

~~Total DDx concentrations in sediments in much of the channel and in most of the Study Area upstream of~~ RM ~~8 are generally less than 5 µg/kg, downstream of~~ RM ~~8, concentrations in the channel are trypcially between 5 and 10 µg/kg. This compares to an upriver background level of XX µg/kg (on OC-equivalent basis).~~

Concentrations are typically greater in the subsurface than in the surface layer, indicating DDx sources are primarily historical. The concentrations of DDx in surface sediments are greater in the Study Area than those in the upriver, downtown, Multnomah Channel, and downstream reaches. When DDx concentrations are averaged on a river-mile basis, the greatest difference between surface and subsurface sediment concentrations is observed in the western nearshore zone between RM 7 and 8, where the highest concentrations are found at depths greater than 10 ft (Figures 5.1-11 and 5.1-12; Panels 10.2-3A–B), indicating greater past loads. ~~Areas of~~ high ~~elevated~~ DDx concentrations ~~are more common~~ found downstream of RM 8 ~~(where concentrations~~ are typically in the 5 to 100 µg/kg range.~~)~~ ~~compared with~~ U~~u~~pstream of this location~~,~~ ~~where~~ concentrations are typically less than 5 µg/kg.

DDx compounds are widely detected in Portland Harbor surface and subsurface sediments; other abiotic media, such as surface water, sediment traps, and TZW; and fish and shellfish tissues. The highest DDx concentrations are found in several nearshore areas associated with known historical DDx sources (Arkema, Rhone Poulenc, and Willbridge Terminal) along the western shore of the river from about RM 6.8 to RM 7.5. Throughout the Study Area, DDx concentrations are generally

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lower in surface than subsurface sediment, indicating greater historical inputs or past releases.

The highest DDx concentrations observed in surface water, sediment traps, TZW, and biota samples are from the area of RM 6.8 to RM7.5. Sediment concentration gradients along this western nearshore area and in the edge of the navigation channel downstream from this source area indicate the downstream transport of DDx in sediments and possible inputs from other point sources. This pattern is consistent with the hydrodynamic and sediment transport characteristics in this portion of the Study Area, particularly from RM 5 to 7. Less extensive areas with elevated DDx occur at Gunderson, at the head of International Slip, RM 4.8W (subsurface only), and at the Cascade General Shipyard at depth in the sediment column.

DDx concentrations in surface water transect samples collected at the upstream end of of the Study Area and upriver (Panel 10.2-3A) in high-flow conditions (Figure 5.3-59) are indicative of inputs from upriver areas. The results of single-point samples collected downstream of RM 7.5, and the transect at RM 6.3 exhibit elevated DDx concentrations in both low-flow and high-flow conditions (Figure 5.3-59; Panel 10.2-3A). Results of surface water samples collected at RM 2 do not exhibit elevated concentrations although results from the surface water transect at the mouth of Multnomah Channel exhibited elevated total DDx concentration during low-flow conditions (Panel 10.2-3A; Figure 5.3-59). The DDx concentrations were higher in unfiltered samples than in filtered samples, combined with the low aqueous solubility of DDx compounds this indicates the DDx is more associated with the solids.

DDx concentrations in the sediment trap samples were generally low throughout most of the Study Area, although higher concentrations were measured in traps at RM 6 and 7.5W and in Swan Island Lagoon in the summer and fall periods. The elevated DDx concentrations reported at RM 11E in the fall period are likely an artifact resulting from analytical interferences associated with PCBs also detected in that sample.

Elevated DDx concentrations were observed in TZW samples collected near RM 7.2W TZW samples from other areas were not analyzed for DDx, precluding comparisons with other areas. The DDx concentrations were higher in unfiltered samples than in filtered samples, combined with the low aqueous solubility of DDx compounds this ~~suggests~~indicates the DDx ~~concentrations were~~is more associated with the solids.

The highest concentrations of DDx in biota are found where sediments concentrations are highest, and most extensive near RM 7 (Figure 5.5-12a-j, Panel 10.2-3C).

10.2.2.7 Potential DDx Sources and Pathways

Within Portland Harbor, DDx was historically associated with pesticide manufacturing and storage facilities. It also was released to air and soil through widespread spraying of crops and forests, and for mosquito control throughout the Willamette River Basin. Some riverbank facilities and other upland properties likely applied DDT for mosquito control before it was banned. Known historical chemical manufacturing and/or storage sites are presented on Map 3.2-5, and include Arkema, Rhone Poulenc, and the Shell Terminal at Willbridge. As shown on Table 10.2-4 and Panels 10.2-3A–C historical known complete or likely complete pathways for DDx have been identified at four sites. Of these, the groundwater pathway is included at two sites, stormwater at four sites, and riverbank erosion at two sites. Historical known complete or likely complete pathways for DDx were not identified for overwater releases or overland runoff. Currently known complete or likely complete pathways for DDx have been identified at three sites and include groundwater migration, stormwater, and riverbank erosion at one site. No current known complete or likely complete pathways were identified for overwater releases or overland runoff.

Areas of DDx contamination are found downstream of RM 7, and may be related, in part, to downstream transport of sediment in the relatively higher energy areas that extends from about RM 7 to RM 5 (see Section 3.1.5). DDx concentrations observed in other upstream and downstream areas are not associated with known sources (Panels 10.2-3A–B).

Historical DDx sources to the Study Area included upstream surface water, stormwater, and riverbank erosion (Table 10.2-4). The historical releases from these pathways are not quantifiable. Overall, current DDx inputs to the Study Area are much lower than historical inputs because DDT is no longer manufactured or used in widespread spraying applications and because waste management practices have greatly improved. However, elevated DDx concentrations in surface sediments and in other media, including biota, at RM 6.8 to 7.5W, and at RM 8.8W indicate localized, external inputs or ongoing internal mass transfer of historical DDx from subsurface/surface sediments to other media by processes such as sediment resuspension and biological uptake.

The most significant current influx of DDx to the Study Area is upstream surface water, while stormwater transport is the most significant current pathway to the Study Area from adjacent upland sites. DDx was detected in stormwater from each land use area sampled, outfalls associated with the heavy industrial land use category and selected individual non-representative outfalls contributed the majority of the estimated DDx stormwater load (see Section 6.1.2.3). Current and/or historical known or likely complete pathways for DDx in stormwater have been identified at Arkema, Rhone Poulenc, Metro Central Transfer Station, the Willbridge Terminal, and City of Portland Outfalls OF-22 and OF-22B. Most of the facilities either drain stormwater or infiltrate contaminated groundwater to shared conveyances (Saltzman Creek, OF-22B, or OF-22). Source control measures taken at the Arkema site have largely eliminated the stormwater pathway from this site. A historical likely complete pathway for DDx in stormwater has also been identified for Willbridge Terminal. No potentially complete current or historical overland transport pathways or historical overwater releases of DDx have been identified for any sites.

Known or likely historical DDx groundwater pathways have been identified at Arkema and Rhone Poulenc. Cleanup efforts conducted by Rhone Poulenc are expected to eliminate the preferential groundwater pathway to stormwater. DDx is present in upland groundwater plumes at the Arkema site, and it has been detected in nearshore wells. Groundwater controls at the Arkema site have greatly reduced the groundwater pathway.

Known or likely complete Historical DDx riverbank erosion pathways have been identified at Arkema and Willbridge Terminal, At Arkema, e the riverbank area received miscellaneous fill that included miscellaneous materials from spent chlorine cells for several years (ERM 2005). The fill material included clean soil, In addition, dredge spoils were deposited on the riverbank. Riverbank erosion at the site has also been identified as a known complete current pathway for DDx.

~~There are no known atmospheric sources of DDx within the Study Area.~~

~~Upriver of the Study Area, DDx was identified as a COI for two sites based on hazardous substance/waste type characterizations. Of these two sites, DDx was detected in sampled media at the OHSU Moody Ave. Units A, B, C (including #1401 SW Moody Right-of-Way) property. Potential pathways to the river include groundwater and stormwater/surface runoff from the site. Upstream DDx sources with pathways potentially impacting surface water may be important since upstream surface water represents the largest current loading term for DDx to the Study Area~~

10.2.2.8 Loading, Fate, and Transport of DDx

Total DDx loading, fate, and transport assessment for the Study Area in a typical year is summarized on Figures 10.2-7a-b through 10.2-9. Estimates are for current conditions, and likely differed historically. Much of the total DDx mass in the Study Area, especially in deeper sediments, is attributable to historical loading that occurred under different loading conditions and rates.

Historical DDx sources to the Study Area included upstream surface water, stormwater, and riverbank erosion (Table 10.2-4). The most significant current influx of DDx to the Study Area is upstream surface water, and is associated with both the dissolved and suspended particulate fraction. The majority of the mass load enters the Study Area during high-flow conditions (Figure 6.1-6), indicating that erosion of upstream agricultural areas is an important upstream source. While additional DDx mass also enters and remains in the Study Area with upstream bedload sediments, it has not been quantified. Total DDx loads in upstream surface water (RM 11) and the downstream boundary of the Study Area (combined loads at RM 2 and the Multnomah Channel entrance) are comparable. Additional DDx loading, in order of importance, are stormwater[6], atmospheric deposition to the river surface (atmospheric deposition to the

[6] The DDx stormwater loading term developed in the RI is dominated by the estimated load from a single "non-representative" site. As discussed in Section 6, the estimated loads from this non-representative site are subject to a high degree of uncertainty related to extrapolation of stormwater concentrations measured in the non-

watershed is included in the stormwater term), groundwater advection through subsurface sediments, and upland groundwater plume discharge.

Internal DDx fate and transport for the Study Area is shown on Figure 10.2-8. DDx transport to the water column due to pore water advection through surface sediments is only process for which quantitative estimates were developed. Other internal fate and transport processes are depicted on Figure 10.2-8 on a qualitative basis only. DDx partitioning between suspended sediment and surface water depends on the relative chemical concentrations associated with suspended particulate organic carbon and the dissolved surface water fraction, as well as reaction kinetics.

DDx loads from permitted non-stormwater point source discharges are not expected to be significant and estimates were not generated.[7] Estimates of DDx loading from upland soil and riverbank erosion also were not assessed in the RI due to a paucity of data for riparian soil DDx concentrations and erosion rates.

Total DDx load in surface water increases moving downstream through the Study Area (Figure 10.2-9), with the largest DDx stormwater inputs entering between RM 6.8 and 7.4. Upland groundwater plume loads are highest between RM 7 and 8. Estimated current DDx annual loads from advection through subsurface and surface sediments follow the patterns of DDx sediment concentrations (Figure 6.1-59), with the highest loads between RM 7 and 8.

Estimates of quantifiable external loading terms indicate that the highest current external inputs to the Study Area are from upstream surface water. Estimates of DDx fluxes in surface water at RM 2, the downstream end of the Study Area, and in Multnomah Channel suggest that slightly more DDx mass may be leaving the Study Area downstream in surface water than entering the Study Area from all quantified sources. This possible increase may reflect an internal mass transfer of DDx from bedded sediment to the water column, likely through sediment resuspension. The relationships between tissue body burdens and abiotic concentrations across the Study Area are a primary focus of the fate and transport modeling to be conducted at the Site for the FS.

10.2.2.9 Human and Ecological Risks Assciated with DDx

DDD, DDE, and DDT ~~were found to~~ pose unacceptable human health risks associated with consumption of resident fish, though this was largely limited to the area at RM 7W. DDx compounds in sediment pose potentially unacceptable risk to the benthic community only on the western side of the river between approximately RM 6.8 and 7.4.

representative outfalls (<10 percent of the drainage sub-basin) to areas that are larger than the catchments that drain to the non-representative outfalls.

[7] NPDES permitted wastewater discharge is not expected to contain significant amounts of total DDx relative to other loading terms.

10.2.3 Total PAHs

The graphical CSM for PAHs is shown on Panels 10.2-4A–C. PAHs are a large chemical group composed of more than 100 chemicals that are constituents of crude and refined oil, shale oil, coal tar, and creosote. PAHs are also formed during the incomplete combustion of organic materials including coal, oil, gas, wood (wood stoves, fireplaces), garbage (municipal waste incineration), or other organic substances, such as tobacco. Natural sources of PAHs include volcanoes and forest fires. The largest historical sources of the high concentrations of PAHs in Study Area sediments were localized releases, such as historical industrial direct discharges, from facilities along the Study Area that formerly or currently use, manufacture, and/or store products and waste materials containing PAHs (petroleum oils and coal-based products, petroleum product use and storage, used oil, and asphalt roofing materials).

PAHs may accumulate in benthic organisms, fish, and other organisms that ingest sediments while feeding. However, biomagnification through the food chain is expected to be minimal as fish, mammals, and birds all possess the ability to metabolize PAH compounds.

10.2.3.1 PAH Contaminant Distribution

PAHs are present at a wide range of concentrations throughout the Study Area in all mediaOn a harbor-wide basis, the highest PAH concentrations in sediments generally occur downstream of RM 7 in nearshore areas proximal to local upland sources (Maps 5.1-9 and 5.1-10a–m; Panels 10.2-4A–B) offshore of Siltronic, Gasco, Marine Finance, and Foss Brix. Elevated PAH concentrations are also observed in surface and subsurface sediments in the navigation channel between RM 4 and 6.5. Other areas of elevated total PAH concentrations in surface sediments include Mar Com South (RM 5.5–5.6E), Terminal 4 Slip 3 and Wheeler Bay (RM 4.3–4.6E), Slip 1 (RM 4.3E), and the International Slip (RM 3.7–3.8E). The portion of the Study Area upstream of RM 7 (not including Swan Island Lagoon) is characterized by widespread PAH concentrations less than 500 µg/kg. In contrast, downstream of RM 7 and away from the high concentration areas associated with known sources, and also in Swan Island Lagoon, PAHs concentrations are generally1,000–5,000 µg/kg. ~~For comparison, the upriver background PAHs level is XX µg/kg (on OC-equivalent basis).~~

Total PAH concentrations are generally higher in subsurface sediments within the Study Area as a whole (Panel 10.2-4A–B; Maps 5.1-10a-m; Figure 5.1-45), pointing to higher historical inputs to the Study Area. The most notable exception to this pattern is the navigation channel at RM 5 to 6.5 where the PAH concentrations in surface sediment are greater (Maps 5.1-10f-g), indicating downstream transport of PAHs in surface sediments once they reach the channel from adjacent or upstream nearshore zones. This is consistent with the relative dynamic channel environment found from RM 5 to 7 (see Section 3.1.5). Other exceptions to the general pattern of higher subsurface total PAHs include Swan Island Lagoon and Multnomah Channel, where higher PAH concentrations are observed in the upper portion of the sediment column and not just the surface layer.

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PAH compositions in sediment trap and high-flow surface water particulate samples were generally similar to that of the sediments, indicative of increased bedded sediment resuspension during higher flow periods. PAH composition in sediment trap and surface water during low-flow and stormwater-influenced surface water particulate samples generally did not correlate well, indicating that current, lateral and upstream sources differ in composition from the PAHs in the bedded sediments. Total PAH concentrations in surface water are elevated predominately during low-flow conditions (Figure 5.3-77), further illustrating the localized nature of the PAH sources in the Study Area. PAH concentrations in sediment trap samples are notably elevated at RM 6W in all seasons sampled (Figure 5.2-19). Less distinct spikes were measured from May to August in Swan Island Lagoon, and from August to November in Multnomah Channel.

Total PAH concentrations in TZW were reported in areas that correspond with elevated areas of sediment concentrations. The highest concentrations reported in biota samples also correspond with areas where PAH concentrations sediment are elevated.

10.2.3.2 Potential PAH Sources and Pathways

PAHs are associated with bulk fuel storage, MGP, ship and automobile scrapping, asphalt roof manufacturing, use of lubricants, hydraulic, and fuel oils in a large number of industrial and commercial operations, and leaks from vehicles and machinery.

Known complete or likely complete historical pathways for PAHs have been identified at 56 sites, and include stormwater (48 sites), groundwater (11 sites), overwater releases (21 sites), overland transport (10 sites), and riverbank erosion (14 sites). Current known complete or likely complete pathways for PAHs have been identified at 30 sites and include stormwater (16 sites), groundwater (9 sites), overwater releases (14 sites), overland transport (3 sites), and riverbank erosion (4 sites). This information is summarized in Table 10.2-5 and on Panels 10.2-4A–C. Known or likely current and historical upland sources that correlate with areas of high concentrations in sediment include the former MGP site at Gasco, dry dock operations at Cascade General in Swan Island Lagoon, the International Slip, MarCom, bulk fuel terminals at ARCO, ExxonMobil, and Kinder Morgan, and historical releases from McCormick and Baxter in Willamette Cove. Based on reported PAH concentrations in sediment near outfalls draining facilities, it is also likely that stormwater/wastewater/overland transport releases occurred concurrently at Burgard Industrial Park (WR-123), Siltronic and Gasco (OF-22C), and Greenway Recycling, PGE-Forest Park, Willbridge Terminal, Front Avenue LP, and Chevron Asphalt (OF-19). Because wastewater discharges in CSO areas are regulated through municipal pretreatment permits, PAHs are not identified as a COI at any sites with a pretreatment permit (see Table 4.4-5).

Current known and likely complete pathways for migration of PAHs in groundwater have been identified at nine facilities, most of which are associated with bulk fuel storage (Table 10.2-5). Overwater releases are likely to have occurred in association with overwater operations such as fuel transfers and spills, drydock and berth operations, overwater maintenance operations, vessel servicing and emissions, ship repair and maintenance activities, direct discharges of PAH contaminants (tar, oil) to the river, releases during product loading/unloading at docks, and tug and barge operations.

Known complete groundwater pathways were identified at the Siltronic and Gasco sites. Potentially complete groundwater migration pathways were identified at Willbridge Terminal, Kinder Morgan Linnton, ARCO, and ExxonMobil. However, results from TZW samples collected offshore of these four sites indicate that the role, if any, of groundwater transport of PAHs is minor and is not significantly influencing TZW and sediment chemistry.

Bank erosion likely played a bigger role historically as well, particularly during construction in places where contaminated sediments or manufacturing material and debris from upland activities were used as fill. Based on limited riverbank sampling, riverbank erosion is a historical known or likely complete pathway for PAHs at 13 sites: Crawford, Gasco, Gunderson, MarCom South Parcel, Marine Finance, Port of Portland Terminal 4, Slip 1 and Slip 3, Premier Edible Oils, Siltronic, Sulzer Bingham, Triangle Park property, Willamette Cove, and the Willbridge Terminal facility.

~~Table 4.5-1 lists ECSI sites upstream of the Study Area (between~~ RM ~~11.8 and the Tualatin River) and associated COIs. Information for these sites is from DEQ ECSI database. Environmental investigation or cleanup of sediments or upland areas have been ordered for some of these properties, suggesting that they could be now, or in the past have been, significant sources of PAHs to the Willamette River. PAHs were identified as COIs for 22 sites based on their hazardous substances/waste types, and PAHs were detected in sampled media at 14 sites.~~

10.2.3.3 Loading, Fate, and Transport of PAHs

Total PAH loading, fate, and transport in the Study Area in a typical year are summarized on Figures 10.2-10a, 10.2-11a, and 10.2-12, relative loads for LPAHs and HPAHs are presented on Figures 10.2-11b and 10.2-11c, respectively. While these loading estimates are for current conditions, much of the total PAH mass in the Study Area is attributable to historical loading.

Advection through subsurface sediments, upstream surface water, and upland groundwater plumes are estimated to contribute comparable (within an order of magnitude) total PAH loads to the Study Area (Figures 10.2-10a and 10.2-11a). Estimated external PAH loads associated with stormwater, atmospheric deposition to the river, and direct discharges from permitted non-stormwater point sources are one to two orders of magnitude lower than the other external terms. LPAHs contribute the bulk of the total PAH load for all loading terms evaluated quantitatively (Figures 10.2-11b and 10.2-11c). The majority of the total PAH mass load from upstream surface water enters the Study Area during high-flow conditions (Figure 6.1-8). Upstream surface water total PAH and LPAH loads are associated primarily with the dissolved fraction, whereas HPAH loads are slightly higher in the particulate than in the dissolved fraction. PAH load in surface water increases moving downstream through the Study Area, particularly downstream of RM 7, in both the particulate and dissolved fractions (Figure 10.2-12) under all flow conditions. PAH loading terms that were assessed qualitatively are also shown on Figures 10.2-11a–c. PAH volatilization from the water column may be significant for LPAHs, although this was not evaluated.

Fate and transport processes internal to the Study Area for total PAHs, LPAHs, and HPAHs are also shown on Figures 10.2-11a, 10.2-11b, and 10.2-11c, respectively. Quantitative estimates were developed only for transport to the water column due to pore water advection through surface sediments, and this term is lower in magnitude than the subsurface advective loading term for total PAHs and LPAHs, and similar in magnitude for HPAHs. Estimates of PAH loading from upland soil and riverbank erosion were not assessed due to a paucity of data for riparian soil PAH concentrations and erosion rates.

The total PAH load in surface water increases moving downstream through the Study Area,[8] particularly downstream of RM 7 (Figure 10.2-12). Most of the load from the two largest external lateral loading terms, subsurface sediment advection and upland groundwater plumes, enters the Study Area between RM 6 and 7.

PAH contamination in sediments is associated with known or likely historical and current sources. Two areas of elevated PAH concentrations, offshore of the Gasco former MGP site at RM 6.5W and the Port's Terminal 4, Slip 3, are the focus of early cleanup actions independant of the harborwide RI/FS. Releases associated with Gasco and other known and potential sources along the west side of the river between RM 6 and 7 have resulted in higher nearshore PAH concentrations in both surface and subsurface sediments. Downstream concentration gradients both nearshore and in the navigation channel in this relatively high-energy portion of the river indicate downstream transport of PAHs extending approximately to the Multnomah Channel entrance. In the navigation channel from RM 5 to 6, high surface sediment concentrations relative to subsurface levels is indicative of transport of material through this reach rather than long-term accumulation. Other, less extensive areas of PAH sediment contamination in the Study Area, are associated with known or likely sources including offshore of Cascade General, in Willamette Cove (subsurface sediments), offshore of Mar Com, at the head of the International Slip, and at several nearshore locations along the west bank from RM 3 to 5.

In general, PAH concentrations in subsurface sediment are greater than in surface sedmiment, indicating greater inputs or releases historically that have been reduced or eliminated over time. Elevated PAH levels in other media—TZW, surface water, sediment traps, and biota (particularly clams)—are largely restricted to the most

[8] At approximately RM 3, the Columbia River and Multnomah Channel hydraulically influence the flow regime complicating interpretation of load conditions in this area (see Section 3.1.3).

extensive area of elevated sediment concentrations, along the western shore around RM 6.

Overall, current PAH inputs to the Study Area are much lower than historical inputs because material handling and waste management practices have greatly improved. Measured elevated concentrations of PAHs in surface sediments and other media including surface water (e.g., Swan Island Lagoon, RM 7.4W, RM 6.8W) and biota in specific areas (e.g., International Slip; Terminal 4, Slips 1 and 3; offshore of Gasco; and Swan Island Lagoon) indicate that localized inputs and/or internal mass transfer of sediment PAHs by processes such as sediment resuspension likely affect other media. Potentially important current pathways from upland sources include groundwater plumes, overwater releases, overland transport, and riverbank erosion.

Emprical estimates of current external PAH loads (mass/yr) to the Study Area indicate that advection through subsurface sediments, upstream surface water, and upland groundwater plumes contribute comparable total PAH loads to the Study Area, whereas loads associated with stormwater, direct atmospheric deposition to the river, and direct discharges from permitted non-stormwater point sources are considerably less important. These data indicate that surface water PAH mass loads increase from upstream to downstream, likely reflecting inputs from the other external loading terms, which peak at RM 6 to 6.9. Internal mass transfer from bedded surface sediments to surface water/biota from sediment resuspension erosion was not quantified but likely also contributes to this pattern.

10.2.3.4 Human and Ecological Risks Assciated with PAHs

PAHs were estimated to pose unacceptable risks to human health based on fish and shellfish consumption and direct contact with sediment exposures~~.~~ ; ~~Both human health and ecological risks from PAHs are geographically limited in extent, reflecting the limited areas of very high sediment levels, and are concentrated in~~generally limited to the area of RM 4W to 6W~~ area~~. As a group, PAHs pose the most widespread ecological risk to the benthic community of the contaminants evaluated,~~PAHs~~ and are strongly associated with the benthic risk area from RM 5.1 to 6.9 on the west side of the river~~,~~. ~~where concentrations are elevated in~~Within this area, PAH concentrations observed in ~~sediment and~~ TZW and surface water pose unacceptable risks to fish, invertabrates, amphibians, and aquatic plants . ~~Negligible risks to receptors other than the benthic community are expected~~Risks to aquatic-dependent birds and mammals are negligible. ~~based on the low magnitude and frequency of exceedances in other areas of the river.~~

10.2.4 Bis(2-ethylhexyl)phthalate

A graphical CSM for BEHP in the Study Area is presented on Panels 10.2-5A–C. Phthalates are manufactured, colorless liquids with little or no odor. The primary sources of phthalate emissions are the industries that manufacture it or use it in productions, such as the chemical industry, the plastics industry, the cosmetic industry, machinery manufacturers, and manufacturers of plywood and millwork. Phthalates are

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commonly added to plastics and paint to make the finished product more flexible (ATSDR 2002c) and are widely used as plasticizers in PVC resin. In addition, phthalates are common components of detergents and carriers in pesticide formulations (Xie et al. 2005). Releases to the environment can occur as direct spills from industrial facilities that manufacture or use these chemicals. More commonly, releases occur by leaching of low volumes of phthalates from the wide variety of products that contain them (ATSDR 2002c). Despite its low vapor pressure, BEHP is ubiquitous in the atmosphere due to its widespread use in plastics. BEHP is present in the atmosphere in both the vapor phase and associated with particulates, is subject to both wet (rain and snow) and dry (wind and settling) deposition on the earth's surface (ATSDR 2002c).

The behavior of BEHP in the Study Area environment is largely defined by its high hydrophobicity (the central estimate of log K_{oc} is 7.4). Due to this hydrophobicity, BEHP has a strong tendency to sorb to solids and organic matter in surface water and in sediment-pore water environments. As such, it is not expected to migrate significantly in groundwater. Because of its low vapor pressure, volatilization is a minor loss mechanism for BEHP, particularly when sorbed to solids. BEHP is subject to fairly rapid degradation in the atmosphere, but much slower abiotic and microbially-mediated degradation processes under aerobic conditions in sediment and surface water (HSDB 2006; ATSDR 2001). While its physical properties indicate that BEHP would be expected to bioconcentrate in aquatic organisms; however, Study Area biota results suggest that BEHP is readily metabolized.

10.2.4.1 BEHP Contaminant Distribution

Elevated BEHP concentrations in sediments were reported, with minor exceptions, in nearshore areas outside the navigation channel and proximal to local upland sources (Maps 5.1-12a–m and Panels 10.2-5A–B), and are observed in Swan Island Lagoon and in the International Slip (RM 3.7–3.8E), and along the riverside of Schnitzer/Calbag site RM 3.8–4.1E. RM 7.6E, RM 9.7W, RM 8.8W, RM 8.3W, RM 7.6W, and offshore of RM 7.1, and RM 10 in the navigation channel. Elevated subsurface concentrations were less widespread, and typically observed in areas with elevated surface concentrations. Exceptions were noted at RM 10.5W and at RM 5.7W.

Elevated concentrations of BEHP were detected in only one surface water sample collected during high-flow conditions, at RM 8.6W . BEHP concentrations in sediment trap samples did not vary widely spatially or temporally throughout and upstream of the Study Area, with the exception of notably elevated concentrations measured in the two samples (summer, fall) collected in Swan Island Lagoon (see Figure 5.2-22).

Because of BEHP's hydrophobic nature, groundwater is unlikely to be a significant historical or current pathway for BEHP migration into the Study Area and was not included in the TZW sampling program. Thus, no data exist to corroborate this hypothesis.

BEHP was detected in laboratory-exposed clams and worms, mussels, and fish. However, it was not detected in crayfish, juvenile Chinook, or carp. The highest reported concentrations in biota were generally detected on both sides of the river near RM 4 and above RM 9.5, and one location at the downstream end of Swan Island. With the exception of the surface sediment on the east bank near RM 4 and subsurface sediment at the downstream end of Swan Island, elevated BEHP concentrations in biota do not correlate well with elevated concentrations in sediment.

10.2.4.2 Potential BEHP Sources and Pathways

Historical known complete or likely complete pathways include stormwater (28 sites), overwater releases (three sites), overland runoff (one site), riverbank erosion (two sites), and groundwater (one site) (Table 10.2-6). Current known complete or likely complete pathways for BEHP have been identified at 16 sites and include groundwater (three sites), stormwater (12 sites), and overwater releases (two sites).

BEHP concentrations are elevated in surface sediment in current and former shipyards such as Swan Island and the International Slip. BEHP is likely to have been released to Swan Island Lagoon and the Portland Shipyard for many years, and continued inputs may occur from known sources and contributions from the numerous outfalls in this area. The lack of elevated concentrations at depth in the sediment column in Swan Island Lagoon may indicate low burial rates, surface sediment mixing, and/or higher recent inputs. Higher concentrations in subsurface sediment near the shipyard docks indicate historically high levels and burial over time.

There are no known sources of BEHP associated with sites that discharge stormwater to the International Slip. However, this area has an auto shredding facility, and BEHP has been identified with metal scrapping (see Section 3.2.1.4). It was also the former location of a large shipyard owned by the Oregon Shipbuilding Corporation. Several metals facilities (either fabrication or scrapping) and a historical paint spill area have been identified as BEHP sources that discharge to OF-19 at RM 8.3W. Identified sources of BEHP at RM 8.8W include Gunderson (outfalls from the Marine Barge Paint and Blast Area) and multiple facilities discharging to OF-18.

BEHP migration by overland transport has been identified at the MarCom South Parcel as a historical likely complete pathway. Current overwater releases may be locally important at sites with continuous waste handling or operational activities, but are considered a minor current pathway overall. Cascade General and Mar Com South represent the only currently known or likely complete BEHP overwater pathways. Groundwater Discharge been identified as a current or historical complete pathway at Premier Edible Oils, Triangle Park property, and the Willbridge Terminal facility. There are no known atmospheric sources of BEHP within the Study Area.

No upriver watershed sources of BEHP have been identified.

10.2.4.3 Loading, Fate, and Transport of BEHP

BEHP loading, fate, and transport assessment for the Study Area in a typical year is presented on Figures 10.2-13a through 10.2-15. The highest relative current external inputs are from upriver surface water,[9] with relatively minor additional contributions from Study Area stormwater and advection through subsurface sediments. The total BEHP loads in surface water upstream and the downstream boundary of the Study Area are generally comparable. Although the current upstream surface water load estimate exceeds the other loading terms, there is no indication that the surface water load is responsible for spatial distribution of the BEHP observed in Study Area sediments, and much of the surface water load appears to pass through the Site. As noted, the off-channel and nearshore elevated BEHP concentrations appear to be associated with localized upland sources and pathways.

A cross-media comparison of surface sediment, sediment trap samples, and particulate suspended solids on a Study Area-wide basis (Tables 10.2-2a-b and Figure 10.2-13b) show that the mean BEHP concentrations differ across all media, the highest average concentrations were observed surface sediment, followed by surface water particulatesl and then sediment trap samples.

10.2.4.4 Human and Ecological Risks Assciated with BEHP

BEHP poses a risk to human health for tribal fishers consuming both resident and migratory fish cought within Portland Harbor. It was identified as posing unacceptable risk to ~~benthic~~ invertebrates, fish, amphibians, and other aquatic receptors. However, based on the frequency of exceedances and generally low magnitude of exological risks for all species except smallmouth bass and sculpin, negligible risks are expected.

10.2.5 Total Chlordanes

A graphical CSM for total chlordanes Study Area is presented on Panels 10.2-6A–C. Chlordane is a manufactured chemical that was used as a pesticide on crops, including corn and citrus, and on home lawns and gardens in the U.S. from 1948 to 1988. It was also used from the 1950s to the 1980s to prevent or eliminate termites. Some of its trade names are Octachlor and Velsicol 1068. Chlordane does not occur naturally in the environment. The EPA banned all uses of chlordane, with the exception of termite control, in 1983; all uses were banned in 1988 (ATSDR 1995). Chlordanes are hydrophobic, and to sorb to solids and organic matter in surface water and sediment. They are persistent in sediments, and subject to very slow abiotic degradation processes. The dissolved fraction in surface water is subject to volatilization. Chlordanes can bioaccumulate in the tissues of fish, birds, and mammals (ATSDR 1995).

[9] The surface water data set only has total concentrations for BEHP; therefore, Figure 10.2-13a presents only the total estimated surface water loading rates for this chemical.

10.2.5.1 Total Chlordanes Contaminant Distribution

Total chlordanes are the sum of oxychlordane, trans-chlordane, cis-chlordane, trans-nonachlor, and cis-nonachlor. Each of these chemicals is analyzed individually and the concentrations are summed to obtain the total chlordanes value.

Several sediment, surface water, and biota samples were reported with high detection limits for total chlordanes, some of which meet or exceed the above definition of elevated (Panels 10.2-6A–C). These high detection limits are the result of chromatographic interferences, laboratory blank contamination, mass spectrometer details related to identification of the components of total chlordanes, or limited sample sizes. High detection limits may obscure the presence of total chlordanes at a concentration below the elevated detection limit, but a high detection limit does not imply the presence of the chemical.

On a harbor-wide basis, the highest detected concentrations chlordane in sediments are restricted to small, widely scattered nearshore or off-channel areas, proximal to local upland sources (Maps 5.1-13 and 5.1-14a–m and Panels 10.2-6A–B). Areas where surface and subsurface sediment concentrations are greater than 10 µg/kg include RM 5.8W-9W, and approximately RM 3E, 4E, 5.5E, and 11E. Total chlordanes were detected in a majority of surface water samples. Elevated surface and subsurface concentrations are also found in Swan Island Lagoon, RM 5.6, and International Slip along the eastern nearshore. Reported concentrations in sediment trap samples were typically low (<4 µg/kg), with no strong temporal or spatial patterns in the measured concentrations (Figure 5.2-23). Chlordanes were detected at low concentrations with varying frequency in all fish and invertebrate samples.

10.2.5.2 Potential Sources and Pathways of Total Chlordanes

The known sources of total chlordanes to the Study Area are summarized in Table 10.2-7 and on Panels 10.2-6A–C. The former Rhone Poulenc pesticide manufacturing facility is the only source of chlordanes currently identified that historically discharged manufacturing waste and stormwater to the river at approximately RM 6.9, and site groundwater infiltrates to City of Portland outfall OF-22B. However, the distribution of chlordane in nearshore sediments is indicative that other sources may also be present.

Historical known pathways for stormwater exist at the Rhone Poulenc facility, including infiltration of contaminated groundwater into the storm system and the City of Portland outfalls, specifically OF-22B (RM 6.9W). Total chlordanes have been detected in upland soils at the former Rhone Poulenc property, and stormwater is a likely complete historical pathway. Historically, manufacturing wastes from Rhone Poulenc was discharged to Doane Lake, which occasionally discharged to the river via a historical drainage ditch from 1972 to 1980. This ditch entered the river near RM 6.9, an area with elevated surface and subsurface sediment total chlordanes concentrations. Currently, site stormwater is collected, treated, and discharged through WR-6, also located at RM 6.9W. No current known or likely complete overland transport pathways for total chlordanes have been identified.

Total chlordanes were detected in sediment samples collected during the PGE Willamette River Sediment Investigation, and chlordanes were likely used as a pesticide in the agricultural areas of the Willamette River watershed up until the late 1980s so upriver sources continue to exist.

10.2.5.3 Loading, Fate, and Transport of Total Chlordanes

The loading, fate, and transport of total chlordanes in the Study Area in a typical year is summarized on Figures 10.2-16a through 10.2-18 . Current external inputs are dominated by upriver surface water, with relatively minor additional contributions from Study Area stormwater, advection through subsurface sediments, and atmospheric deposition to the river surface. The total chlordane upstream surface water loads are comparable with the combined loads at RM 2 and the Multnomah Channel entrance in both the dissolved and particulate fractions. Although the current upstream surface water load estimate exceeds the other current loading terms, there is no indication that this load is responsible for distribution of total chlordanes in Study Area sediments.

Cross-media comparisons of surface sediment, sediment traps, and suspended solids in surface water (Tables 10.2-2a-b and Figure 10.2-16b) show that the overall surface sediment concentrations are greater than those in surface water particulate and sediment trap samples. Surface water particulate and sediment trap concentrations are similar.

10.2.5.4 Human and Ecological Risks Assciated with Chlordanes

Total chlordanes are estimated to pose a cancer risk greater than the lower end of EPA's risk management range based on a harbor wide consumption of resident fish at a subsistence level. Chlordanes in sediment were identified as posing ~~unacceptable~~ecologically significant risks to benthic invertaebrates, ~~They were not identified as a contaminant posing unacceptable risks to ecological receptors.~~

10.2.6 Aldrin and Dieldrin

Graphical Study Area CSMs for aldrin and dieldrin are presented on Panels 10.2-7A–C and 10.2-8A–C, respectively. Aldrin and dieldrin are organochlorine insecticides with similar chemical structures, manufactured for agricultural use from the 1950s to 1987. From the 1950s until 1970, aldrin and dieldrin were widely used insecticides for crops and livestock. Peak production occurred in the mid-1960s. Aldrin/dieldrin ranked second—after DDT—among agricultural insecticides used in the U.S. in the 1960s (Jorgenson 2001). EPA banned the use of aldrin and dieldrin in 1974, except to control termites. By 1987, EPA banned all uses (ATSDR 2002b).

Like the other pesticides on the CSM chemical list, aldrin and dieldrin are ~~highly~~ hydrophobic and have a strong tendency to sorb to solids and organic matter in surface water and sediment. Aldrin is subject to abiotic and microbially-mediated degradation processes as well as photolysis in surface water. Dieldrin is more recalcitrant, subject to

Commented [AE14]: Another one of those subjective modifiers very slow degradation processes in the environment. Both chemicals are subject to fairly slow volatilization (ATSDR 2002b). Plants can absorb both aldrin and dieldrin from the soil. Dieldrin can bioaccumulate, while Aldrin does not as it is quickly metabolized to dieldrin in plants and animals (ATSDR 2002b).

10.2.6.1 Aldrin and Dieldrin Contaminant Distribution

A number of sediment, surface water, and biota samples were reported with high detection limits for aldrin or dieldrin as a result of chromatographic interferences, laboratory blank contamination, mass spectrometer criteria issues, or limited sample ~~sizes~~~~mass~~volume. High detection limits may obscure the presence of aldrin or dieldrin at a concentration below the detection limit, but a high detection limit does not imply the presence of the chemical.

Aldrin and dieldrin contamination in sediment is generally restricted to small, widely scattered nearshore areas. Elevated concentrations were generally not detected in off-channel or navigation channel areas, with the exceptions of detections of dieldrin at RM 11.4E (which includes several samples nearshore and offshore with elevated concentrations) and the head of the International Slip (RM 3.7E). Surface and subsurface sediments results show elevated concentrations of aldrin and dieldrin at RM 6.8 to7.5W and RM 8.8W. Overall, aldrin concentrations slightly higher in subsurface sediments, while dieldrin concentrations are generally higher in the surface sediments. Areas with high concentrations of aldrin and dieldrin are generally co-located.

Aldrin and dieldrin were detected both in particulate and dissolved surface water samples, with the dissolved fraction slightly to largely predominating with a major exception at RM 6.9. Aldrin and dieldrin were infrequently detected in sediment trap, most frequently downstream of RM 8. Tissue concentrations of aldrin and dieldrin concentrations were generally less than 10 µg/kg, although higher dieldrin concentrations were observed in sculpin collected from RM 2E- to RM 4E.

10.2.6.2 Potential Aldrin/Dieldrin Sources and Pathways

The known sources of aldrin are summarized in Tables 10.2-8 and 10.2-9 for aldrin and dieldrin, respectively. The only currently identified source of aldrin and dieldrin within the Study Area is the former Rhone Poulenc pesticide manufacturing facility that historically discharged manufacturing waste and stormwater to the river at approximately RM 6.9. Aldrin and dieldrin have been detected in upland soils at this site, and stormwater is a known complete current and historical pathway. However, based on the distribution of aldrin and dieldrin in nearshore sediments, other sources may be present.

Historically, manufacturing wastes from Rhone Poulenc were routed to Doane Lake, which occasionally discharged to the river via a drainage ditch near RM 6.9 from 1972 to 1980. Currently site stormwater is collected, treated, and discharged through WR-6, also located at RM 6.9W. No current known or likely complete overwater or riverbank erosion pathways for aldrin or dieldrin have been identified. Atmospheric deposition is a potential historical and current pathway, but has not been quantified.

Jorgenson (2001) lists Van Waters & Rogers (Univar), as a producer and/or distributor of aldrin and dieldrin. However, there is no information about releases to the environment from this facility.

10.2.6.3 Loading, Fate, and Transport of Aldrin and Dieldrin

The loading, fate, and transport assessment of aldrin in the Study Area for a typical year is summarized on Figures 10.2-19a through 10.2-21, and on Figures 10.2-22a through 10.2-24 for dieldrin. Upriver surface water is the largest estimated external loading term, surface water loading of dieldrin is approximately 50 times than that of aldrin. Stormwater is the second highest estimated loading term, followed by atmospheric deposition to the river surface and advection through subsurface sediments. There is an apparent increase in aldrin loads in surface water between the upstream and downstream boundaries of the Study Area. Conversely, there is little apparent change in surface water dieldrin loads between upstream and downstream boundaries. Although the current upstream surface water load estimate exceeds the other loading terms, there is no indication that the surface water load is responsible for the spatial distribution of aldrin and dieldrin in Study Area sediments, and nearshore aldrin and dieldrin concentrations appear to be associated with localized upland sources and pathways.

Cross-media comparisons of surface sediments, sediment trap samples, and suspended solids in surface water for aldrin and dieldrin are provided in Tables 10.2-2a-b (summary statistics and statistical comparisons) and Figures 10.2-19b and 10.2-22b (box-whisker distribution plots). Aldrin surface sediment and sediment trap concentrations are not significantly different; concentrations in both of these media are greater than the concentration in surface water particles on a Study Area-wide basis. Study Area-wide surface sediment concentrations are significantly greater than the concentrations in surface water particulates, but there appear to be no significant differences between sediment trap concentrations and sediments or surface water particulates.

10.2.6.4 Human and Ecological Risks Assciated with Aldrin and Dieldrin

Dieldrin poses unacceptable human health risks based on consumption of resident fish on both a harbor-wide and localized scale. Dieldrin was identified as an ecologically significant contaminant, primarily due to its ability to biomagnify in the food web. It was infrequently identified as posing unacceptable ecological risks to benthic species. Aldrin, which is rapidly transformed into dieldrin by most fish and wildlife species, was identified as posing potentially unacceptable ecological risks to spotted sandpiper at a limited spatial extent within Portland~~Aldrin and dieldrin were identified as posing unacceptable risks to the benthic community and to sandpipers~~.

10.2.7 Arsenic, Copper, and Zinc

Graphical Study Area-wide CSMs are presented for arsenic, copper, and zinc on Panels 10.2-9A–C, 10.2-10A–C, and 10.2-11A–C, respectively. All three metals are abundant elements in the earth's crust, and natural releases to environmental media can be significant.

Arsenic is a naturally occurring metal that is found widely in natural minerals, including realgar ($As_4S_{4(s)}$), orpiment ($As_2S_{3(s)}$), and arsenolite (As_2O_3; ATSDR 2005a). It occurs naturally in soil, water, and air as a result of mineral weathering, leaching, volcanic eruptions, and wind-blown dirt (ATSDR 2005). Anthropogenic activities, including smelting, use in pesticides, combustion of wood and coal, waste incineration, and the production and use of treated wood products that utilize soluble chromium copper arsenate (CCA), can also release arsenic into the air, soil, water, and sediments.

Arsenic is a redox-sensitive species, existing at the +3 and +5 oxidation states in aqueous environmental conditions. Under oxidizing conditions the As(V) species (H_3AsO_4, $H_2AsO_4^-$, $H_2AsO_4^-$, $HAsO_4^{2-}$, AsO_4^{3-}) predominate, while under reducing conditions the As(III) species (H_3AsO_3, $H_2AsO_3^-$, $HAsO_3^{2-}$, AsO_3^{3-}) predominate (EPRI 1984). Arsenic is generally highly soluble, with few mineral phases exerting controls on aqueous arsenic concentrations under typical environmental conditions. Arsenic sulfide minerals, such as orpiment and realgar, can be formed under reducing and acidic conditions. Although arsenic minerals are generally highly soluble, adsorption reactions to sediment/aquifer mineral grain surfaces frequently limit dissolved arsenic concentrations (Kabata-Pendias and Pendias 1992). Arsenic is particularly strongly adsorbed to iron oxide minerals, with the As(V) species having a greater affinity for the oxide surface than the As(III) species. Because arsenic is frequently present as an anion under typical environmental conditions, its sorption to oxide surfaces is favored at pH <9 (Stumm 1992).

In aquatic environments, bioaccumulation of arsenic occurs primarily in algae and lower invertebrates (ATSDR 2005a). Fish and shellfish can also accumulate arsenic, mainly in the exoskeleton of invertebrates and in the livers of fish. While biomagnification in aquatic food chains is not generally considered significant, predatory fish may biomagnify arsenic through the consumption of prey species (especially bottom dwellers) (ATSDR 2005a).

Copper is an abundant metal element in the earth's crust. Natural releases to environmental media can be significant. Mining operations, agriculture, wastewater sludge, municipal and industrial solid waste, and other industrial processes can also result in environmental releases of copper (ATSDR 2004).

Copper exists in four oxidation states: Cu^0, Cu^{+1}, Cu^{+2}, and Cu^{+3} (Eisler 1998). Of these oxidation states, the cupric ion (Cu^{+2}) is the most likely to be present in water, though rarely as a free ion. As free ions, cupric ions are the most readily available and toxic inorganic species of copper. However, the cupric ions have a strong tendency to complex or sorb to numerous compounds normally found in natural waters, including suspended solids surfaces and dissolved or particulate organic carbon. Such complexation reduces bioavailability to aquatic organisms (Eisler 1998; EPA 2000a). The amounts of the various copper compounds and complexes present in solution in freshwater depend on water pH, temperature, hardness, and alkalinity; concentrations of bicarbonate, sulfide, and organic ligands; size and density of suspended materials; and rates of coagulation and sedimentation of particulates. Up to 29 different species of copper can be present in aqueous solution in the pH range from 6 to 9. The majority of copper in freshwater from pH 6.0 to 9.3 is in the form of carbonate species ($CuHCO_3^+$, $CuCO_3$, $Cu[CO3]_2^{-2}$), which have low toxicity (Eisler 1998). Cupric ions account for less than 1 percent of the total dissolved copper in freshwater. Copper carbonate, cupric hydroxide, cupric oxide, and cupric sulfide will precipitate from solution or form colloidal suspensions when excess cupric ions are present (Eisler 1998). The majority of copper released to surface waters settles out or sorbs to sediments (Eisler 1998). While copper can transform in response to environmental chemistry, it does not degrade.

Copper is taken up by aquatic organisms primarily through dietary exposure and is an essential micronutrient for animals as a component of a number of essential enzymes. Most organisms retain only a small proportion of the copper ingested with their diet. Copper bioconcentrates in aquatic organisms but does not bioaccumulate in mammals or biomagnify in aquatic food chains (EPA 2000a).

Zinc is a common element in the earth’s crust and is released to the environment from both natural and anthropogenic sources. Mining and metallurgical processing are the primary anthropogenic sources, along with use of commercial products such as fertilizers and wood preservatives that contain zinc (ATSDR 1997b). Zinc is also used in galvanizing steel and in soldering formulas.

In the environment, zinc occurs as a sulfide, oxide, or carbonate. In freshwater, zinc is most soluble at low pH and low alkalinity: 10 mg Zn/L of solution at pH 6 that declines to 6.5 mg Zn/L at pH 7, 0.65 mg Zn/L at pH 8, and 0.01 mg Zn/L at pH 9 (Eisler 1993). Zinc in the water column can partition to dissolved and particulate organic carbon. Water hardness (i.e., calcium concentration), pH, and metal speciation are important factors in controlling the water column concentrations of zinc because the divalent zinc ion is believed to be responsible for observed biological effects (EPA 2000a). Because zinc ligands are soluble in neutral and acidic solutions, zinc is readily transported in most natural waters (Eisler 1993). However, most of the zinc introduced into aquatic environments eventually is partitioned into the sediments (Eisler 1993). Zinc release from sediments is enhanced under conditions of high dissolved oxygen, low salinity, and low pH (Eisler 1993). Zinc may change forms in the environment, but it does not degrade.

Zinc is an essential trace element for all living organisms. As a constituent of more than 200 metalloenzymes and other metabolic compounds, zinc ensures stability of biological molecules such as DNA and of biological structures such as membranes and ribosomes (Eisler 1993). Most studies reviewed contained data that suggest that zinc is not a highly mobile element in aquatic food webs, and there appears to be little evidence to support the general occurrence of biomagnification of zinc within marine or freshwater food webs (EPA 2000a). Bioavailability of zinc in sediments is controlled by the acid-volatile sediment concentration. The arsenic- copper-, zinc-related risks to humans and ecological receptors were evaluated in the BHHRA (Section 8 and Appendix F) and BERA (Section 9 and Appendix G).

10.2.7.1 Arsenic, Copper, and Zinc Contaminant Distribution

With the exception of a broad area of relatively elevated copper and zinc concentrations in the vicinity of Swan Island Lagoon and zinc at Terminal 4, Slip 3, elevated arsenic, copper, and zinc concentrations in surface and subsurface sediment are generally restricted to small, widely scattered nearshore areas. The similarity of surface and subsurface concentrations in these areas suggests both recent and historical inputs of all three metals.

Total arsenic, copper, and zinc concentrations in surface water were generally consistent across the entire Study Area. Concentrations were generally higher in low-flow sampling events, and there is generally no relationship evident between elevated surface water and elevated surface sediment concentrations. Sediments collected in and upstream of Study Area over the course of a year in sediment traps show little spatial or temporal trends in measured concentrations. The highest reported arsenic concentrations TZW are located at the west side of the channel at RM 6.2–6.6, and the west bank at RM 7.7. However, there are no corresponding high arsenic concentrations in sediment. The highest copper and zinc concentrations in TZW were measured offshore of the Gasco and Siltronic sites in areas where no elevated surface sediment concentrations were reported. Arsenic, copper and zinc were detected in nearly all fish and invertebrate species and tissues analyzed from within the Study Area.

10.2.7.2 Potential Sources of Arsenic, Copper, and Zinc

The areas of elevated sediment concentrations correspond to the locations of former shipyards, wood treatment facilities, metal recycling operations, pipe manufacturing, metal plating operations, and marine repair facilities. Metals are also associated with some facilities where metal slag and sandblast grit were used as fill. The primary industries in Portland Harbor known to have handled, manufactured, or disposed of arsenic, copper, or zinc include pesticide manufacturing, shipbuilding/demolition and marine repair facilities, metals recycling, battery scrapping, wood treating, and MGP. However, a number of sites with known or likely complete pathways do not appear associated with proximal surface sediment contamination.

Known complete or likely complete historical pathways for arsenic, copper, and zinc have been identified at up to 46 sites.

Although one or more of these metals has been identified as a stormwater COI at sites that drain to a number of municipal and non-municipal shared conveyance systems, associated sediment concentrations are not present near all these outfalls (Tables 10.2-10 through 10.2-12).

Zinc is elevated in sediment in Balch Creek Cove, which is the discharge location for OF-16, OF-17, WR-258, and WR-235. Metals have not been identified as stormwater COIs at WR-258 (drains Fire Station 6) and WR-235 (Port of Portland Terminal 2). GE Decommissioning has a known historical and likely current complete stormwater pathway for zinc. Calbag Nicolai and Galvanizers have likely historical and known current complete stormwater pathways for zinc. All three of these sites are located within the basins of these outfalls.

Arsenic, copper, and zinc concentrations are elevated in sediments offshore of the areas adjacent to Gunderson and the small cove adjacent to the Shaver Transportation and Front Ave LP properties. Gunderson has known complete pathways, both historical and current, for stormwater and overland transport, and Front Avenue LP has a historical known complete stormwater pathway for these metals. Two shared conveyance systems also drain to this area, OF-18 and OF-19. However, none of these metals are elevated in the cove that OF-18 discharges to and only copper is elevated in the vicinity of the OF-19 discharge. Front Avenue LP has a historical likely complete pathway for copper in stormwater and Chevron has a historical known complete pathway for copper in stormwater. Both of these sites drain at least in part to OF-19 (Table 10.2-11).

Elevated sediment concentrations of copper and zinc are present on the nearshore areas on the west side of Swan Island Lagoon and the north end of Swan Island. Cascade General, Swan Island Upland Facility, and Fred Devine are identified as a having likely historically complete stormwater pathway for copper and zinc. Single samples of elevated concentrations of zinc and copper are also present near OFM-1and of zinc near OFM-2. Freightliner TMP and TMP2 drain to these basins and are identified as a having likely historically complete stormwater pathway for copper and zinc.

Stormwater and groundwater infiltration from sites draining to OF-22B and OF-22C (RM 6.8W) are the likely sources of elevated arsenic concentrations in surface sediment at this location. Stormwater has been identified as a historical and/or current known complete pathway at Rhone Poulenc, Metro Transfer Station, Schnitzer-Doane Lake, Gasco, Siltronic, and Gould, which drained to or have had groundwater infiltration to these outfalls.

TZW sampling results indicate that groundwater is not a significant source to sediments. Low subsurface sediment arsenic concentrations may reflect the dynamic setting—arsenic may be transported downstream before there is an opportunity for long-term burial. Downstream surface sediment concentrations are low, suggesting that if this is the case, that the mass of arsenic being discharged is relatively low and is readily dispersed.

Overland transport has been identified as a likely complete historical pathway for both MarCom parcels, as well as a likely complete current pathway for the North parcel. Elevated surface sediment concentrations are located offshore of the MarCom facilities and surface water concentration immediately upstream appears to support stormwater as a pathway. Calbag Metals, adjacent to the International Slip, has been identified as having known current and historical complete stormwater pathways for zinc and copper; Terminal 4 has a likely historical complete overland pathway for zinc.

Arsenic, copper and zinc are associated with Marine Barge Paint and Blast Area operations at Gunderson and a known historical and current complete pathway for groundwater has been identified for arsenic at Gunderson.

Groundwater is a known or likely current and historically complete pathway for these metals for the Siltronic and Gasco sites. However, none of these metals are present at elevated concentrations in surface or subsurface sediment.

Five sites have been identified as having historical known complete pathways for copper and zinc: Gunderson, Cascade General, Swan Island Upland Facility, McCormick & Baxter, and Schnitzer – Calbag. McCormick & Baxter also has a historical complete pathway for arsenic. All but McCormick & Baxter have known or likely current complete overwater pathways. McCormick & Baxter is the only identified site with a known historical complete pathway for arsenic.

Based on limited riverbank sampling, relatively large areas of elevated metals concentrations in surface sediment are associated with sites with known or likely historical complete riverbank erosion pathways for arsenic, copper, and zinc. These sites include Gunderson, Willamette Cove, MarCom South, and McCormick & Baxter. A current known or likely complete pathway for riverbank erosion is limited to Gunderson.

~~There are no known current or historical point sources of atmospheric deposition of these metals in the Study Area. Table 4.5-1 lists upstream ECSI sites (between~~ RM ~~11.8 and the Tualatin River) and associated COIs.~~

~~10.2.7.4~~10.2.7.3 Loading, Fate, and Transport of Arsenic, Copper and Zinc

The loading, fate, and transport of arsenic in the Study Area is summarized on Figures 10.2-25a through 10.2-27. Copper is summarized on Figures 10.2-28a through 10.2-30, and zinc is summarized on Figures 10.2-31a through 10.2-33. Estimated loads from upriver surface water dominate the current loads for all three metals, frequently exceeding the other quantified external loading terms by 2 or more orders of magnitude. Upstream and downstream surface water arsenic loads are comparable. Estimated copper and zinc loads show slightly more entering the Study Area than leaving. The distribution of total arsenic, copper, and zinc (respectively) in surface sediments, sediment trap samples, and the surface water particulate fraction are presented on

Figures 10-2.25b, 10.2-28b, and 10.2-31b, summary statistics in each media are presented in Table 10.2-2a and the results of cross-media statistical testing are provided Table 10.2-2b. For each of these metals, all the distributions of each media are significantly different from each other. Concentrations of each metal in surface water suspended particles are greater than in sediment trap or surface sediments on a Study Area-wide basis. Based on median values, sediment trap concentrations are slighter greater overall than surface sediment concentrations for all three metals.

Although the current upstream surface water load estimate greatly exceeds the other current loading terms, there is no indication that this load is responsible for the spatial distribution of the concentrations of these metals observed in Study Area sediments.

~~10.2.7.5~~10.2.7.4 Human and Ecological Risks Assciated with Arsenic, Copper, and Zinc

Arsenic concentrations in sediment pose unacceptable human health risks due to consumption of fish and shellfish, direct exposure to in-water sediment, direct exposure to beach sediment, and use of the Willamette as a drinking water source~~, and hypothetical use of untreated surface water as a domestic water source~~. Arsenic~~, copper,~~ and zinc were identified as infrequently posing ~~potentially~~ low unacceptable ~~ecological~~ risks to benthic invertebrates. Unacceptable ecological risks from copper were indentified for more lines of evidence than any other contaminant except for PCBs. Unacceptable risks from copper were identified for multiple benthic invertebrate and fish species, as well as to spotted sandpiper. Copper and zinc were identified as ecologically significant contaminants.

10.2.8 Chromium

The Study Area graphical CSM for chromium is presented on Panels 10.2-12A–C. Chromium is a naturally occurring element found in rocks, animals, plants, and soil. It can exist in several different forms in the soil, sediment, water, and air. Chromium(III) occurs naturally in the environment but is also a product of industry. Chromium(0) is used for making steel. Chromium(III) and chromium(VI) forms are produced by the chemical industry and are used for chrome plating, the manufacture of dyes and pigments, leather tanning, and wood preserving. Smaller amounts are used in drilling muds, rust and corrosion inhibitors, textiles, and toner for copying machines (ATSDR 2008).

In the environment, chromium can be found in air, soil, and water. Chromium compounds will usually remain in the air for less than 10 days, depositing to the land and water, especially by wet deposition (ATSDR 2008). Most chromium in water binds to soil and other materials and is subsequently subject to sediment transport processes, though a small amount may dissolve in the water. It can easily change from one form to another in water and soil, depending on the conditions present (ATSDR 2008). The relation between Cr(III) and Cr(VI) in the environment is strongly dependent on pH and oxidative properties of the location, but in most cases the Cr(III) is the dominating species (Kotas and Stasicka 2000).

Although chromium(III) is required in trace amounts for sugar and lipid metabolism in humans and its deficiency may cause a disease called chromium deficiency, chromium(VI) is a toxin and a carcinogen (ATSDR 2008). Fish do not significantly accumulate chromium in their bodies from water (ATSDR 2008).

10.2.8.1 Chromium Contaminant Distribution

Areas of elevated chromium concentrations in surface and subsurface sediments in the Study Area all occur in a few, widely scattered nearshore areas and the head of the International Slip and are limited in spatial extent, and include RM 2E, RM 4E, RM 6E, Swan Island Lagoon, RM 6W, RM 7W, and RM 9W. The distribution of concentrations in surface and subsurface sediments suggest both recent and historical sources.

Sediment trap samples show a uniform distribution of chromium levels seasonally and throughout and upstream of the Study Area (see Figure 5.2-27). Most sediment trap samples fall between 30 and 40 mg/kg; the single highest value (60 mg/kg) was measured in upstream of the Study Area at RM 15.7E during the May to August period.

Chromium was sampled offshore of sites between RM 6.2W and 7.6W. Elevated TZW concentrations (greater than 100 µg/L) were observed between RM 6.2W and 6.5W, offshore of the Gasco and Siltronic properties.

Chromium was detected in all fish and invertebrate species and tissues analyzed within the Study Area.

10.2.8.2 Potential Sources of Chromium and Pathways

In Portland Harbor, the primary industries known to have handled, manufactured, or disposed of chromium include the steel industry (EOSM), ship building/demolition and marine repair facilities (Gunderson, Cascade General, MarCom), metal recycling (Schnitzer-Calbag, Calbag-Nicolai, Calbag Metals-Front Ave., and former operations at Schnitzer-Doane Lake), wood treating (McCormack & Baxter), and MGP sites (Gasco), and heavy oil facilities (bulk fuel and asphalt storage). Sodium bichromate was used in the sodium chlorate manufacturing process as a corrosion inhibitor at Arkema. Historical known complete or likely complete pathways for chromium have been identified at 43 sites (Table 10.2-13 and Panels 10.2-12A–C), and include stormwater (38 sites), groundwater (seven sites), overwater releases (five sites), overland transport (nine sites), and riverbank erosion (10 sites). Current known complete or likely complete pathways for chromium have been identified at 20 sites and include stormwater (13 sites), groundwater (five sites), overwater releases (four sites), overland transport (three sites), and riverbank erosion (three sites).

Groundwater is a historical known or likely complete pathway for chromium at seven sites and a current pathway for five sites. Chromium concentrations near OF-22B and OF-22C may be related to sites with known or likely complete groundwater / stormwater infiltration draining to these outfalls.

Based on limited riverbank sampling, riverbank erosion is a historical known or likely complete pathway for chromium at 10 sites, and a current source at four sites. Chromium concentrations in sediment are indicative of potential releases at EOSM, Schnitzer-Calbag metals, Portland Shipyards, and Gunderson.

Chromium was identified as a COI at 17 upstream sites based on their hazardous substances/waste types. Of these 17 sites, chromium was detected in sampled media at the following five sites:

- Willamette Falls Locks – Chromium detected in upland soil, and direct releases/spills represent potential pathways to the river
- Zidell Marine Corporation – Chromium detected in upland soil, and direct releases/spills, groundwater, and stormwater represent potential pathways to the river
- OHSU - Moody Ave. Units A, B, C – Chromium detected in upland soils, and stormwater and groundwater represent potential pathways to the river
- Clackamette Cove Area – Chromium detected in upland soils, and groundwater represents a potential pathway to the river
- South Waterfront Redevelopment Area 3 – Chromium detected in groundwater, which represents a potential pathway to the river.

10.2.8.3 Loading, Fate, and Transport of Chromium

Chromium loading, fate, and transport assessment for the Study Area is summarized on Figures 10.2-34a through 10.2-36. Estimated loads from upriver surface water dominate the current loads, exceeding the next highest external loading terms—stormwater and upland groundwater plumes—by more than 2 orders of magnitude. The chromium loads in upstream surface water and the downstream boundary of the Study Area are generally comparable, reflecting the absence of significant loads within the Study Area. The nearshore and off-channel areas of elevated chromium concentrations generally appear to be associated with localized upland sources and pathways.

The cross-media comparison of surface sediments, sediment traps, and suspended solids in surface water (Table 10.2-2a-b and Figure 10.2-34b) show that the Study Area-wide concentrations of all media are statistically different, with surface sediment having the highest concentrations, followed by sediment traps and then by suspended solids in surface water.

Within the Study Area, numerous historical and current sources of chromium have been identified for all pathways, but primarily through stormwater discharge. The areas of elevated sediment concentrations generally correspond to the locations of current or former shipyards, wood treatment facilities, pesticide manufacturing, metal recycling operations, steel manufacturing, metal plating operations, and marine repair facilities. Chromium is also identified at some facilities where metal slag and sandblast grit were used as fill. Known and likely current and historical sources of chromium to the Study Area are summarized in Table 10.2-13 and Panels 10.2-12A–C.

The chromium loading estimates indicate that loads from upriver surface water dominate the current loads, exceeding the next highest external loading terms—stormwater and upland groundwater plumes—by more than 2 orders of magnitude. However, the estimated loads entering and leaving the Study Area are comparable, suggesting the absence of significant loads within the Study Area.

10.2.8.4 Human and Ecological Risks Assciated with Chromium

Chromium is estimated to posing unacceptable risk to benthic invertebrates, although the risk of low magnitude and the areal extent is limited.

10.2.9 Tributyltin Ion

A graphical CSM for TBT is presented on Panels 10.2-13A–C. TBT is an organotin compound, and since the mid-1970s has been and is still used as an antifouling agent in paints on the immersed portions of boats and floating structures (Batt 2004). Antifouling paints represent the largest source of TBT in coastal environments, and nationally. Many countries restricted the use of antifouling paints based on the risks to shellfish. The U.S. partially banned the use of TBT-based antifouling paints in 1988 (Showalter and Savarese 2005). Use of TBT compounds as slimicides on masonry, disinfectants, and biocides for various industrial processes also may result in their release to the environment.

TBT is an ionic organic compound, and its partitioning behavior is affected by pH and the identity of anions in solution that pair with the TBT ion (Arnold et al. 1997). Specifically, for pH 10 to 7 the measured log K_{oc} values are on the order of 4; from pH 7 to pH 3 they drop to roughly 2. The mean surface water pH in Portland Harbor is 7.4 (10^{th} percentile is 7.0 and 90^{th} percentile is 7.8). Observed pore water pH values ranged from 5.6 to 8.1. TBT in its nonionic form (not likely observed at the site) is highly hydrophobic, with K_{ow} values on the order of 7(K_{ow}WIN[10]).

Degradation of organotin compounds involves the breaking of the tin-carbon bond. TBT is subject to different degradation mechanisms depending on its location in the environment. In surface water, it is subject to fairly rapid[11] photodegradation and biodegradation (TBT can be degraded by microbial, microalgal, and fungal populations, as well as by some higher organisms, such as fish; Anderson et al. 2002). Degradation

[10] K_{ow}WIN software available online as part of EPA Estimation Program Interface (EPA) suite of programs: http://www.epa.gov/oppt/exposure/pubs/episuite.htm

[11] Half-life information for TBT in freshwater could not be found; however, the following information was found for seawater: the half-life of tributyltin in seawater varies, depending on pH, temperature, turbidity, and light; it is generally estimated to be in the range of 1 day to a few weeks (Alzieu 1998). Biodegradation is the major process in seawaters rich in suspended solids, but photolysis, in surface waters, exceeds biodegradation in clean seawater. Calculated half-lives range from 6 days in summertime waters rich in suspended particles to 127 days in clean winter waters (Watanabe et al. 1992).

of organotin compounds in sediments is much slower than in water, and half-lives have been estimated to be several years (Alzieu 1998). Abiotic cleavage of the tin-carbon bond by hydrolysis is not an important fate process under environmental conditions (WHO 1990).

While accumulation of TBT from water into organisms may be a significant process that can result in elevated tissue concentrations, subsequent biomagnification through the food web is reportedly minor (ATSDR 2005b).

10.2.9.1 TBT Contaminant Distribution

TBT contamination is sediment is primarily located in the vicinity of the Cascade General Shipyard and adjacent to Swan Island Lagoon. Concentrations in subsurface sediments exhibit slightly higher concentrations than surface sediments, suggesting that contributions from historical inputs were greater relative to current inputs. Upstream of RM 7.5 TBT was detected in sediment trap samples only in Swan Island Lagoon.

10.2.9.2 Potential Sources and Pathways of TBT

Within the Study Area, historical and current sources contributed TBT to the river primarily through the overwater and stormwater pathways. Areas of elevated TBT concentrations correspond with current and former shipyards, where ship hull washing, abrasive blasting, and painting occurred in dry docks and berths (see Map 3.2-3).

Identified known and likely current and historical sources of TBT to the Study Area are summarized in Table 10.2-14 and Panels 10.2-13A–C. Historical pathways for TBT migration are found at four current and historical shipyard facilities (Cascade General, Gunderson, Mar Com South, and Marine Finance) and include stormwater (four sites) and overwater, overland, and riverbank erosion (two sites). Current known complete or likely complete pathways for TBT have been identified at two sites (Cascade General and Gunderson).

A current and historical known complete stormwater pathway to the river exists at Gunderson, an active manufacturer and refurbisher of railroad cars and marine barges since 1913 (Integral 2007a). Historical likely complete stormwater pathways also exist at Cascade General, MarCom South, and Marine Finance. Historical overland transport pathways are likely complete at MarCom South and Marine Finance. Overwater discharge is considered to be the greatest contributor of TBT to the river, current and historical complete pathways for overwater discharge of TBT are found at Cascade General and a complete historical overwater patheway existed at MarCom South. There are no current or historical groundwater pathways for TBT in the Study Area. Likely complete historical pathways for riverbank erosion have been identified at MarCom South and Marine Finance. Information on atmospheric deposition of TBT is very limited, but long-range atmospheric transport of butyltins does occur.

TBT was identified as a COI at two upstream sites, Zidell Marine Corporation site (RM 14W) and Ross Island Sand & Gravel (RM 14.7E), based on detections in sampled media.

10.2.9.3 Loading, Fate and Transport of TBT

The loading, fate, and transport assessment for TBT in the Study Area is summarized on Figures 10.2-37a through 10.2-39. Estimated TBT inputs from upriver surface water (11 kg/yr) are lower than those from advection through subsurface sediments (36 kg/yr). The relatively high overall subsurface advective load estimate is driven by localized TBT concentrations at RM 8 to 8.9. Quantitative estimates of internal fate and transport processes were developed only for advection through surface sediments (9.8 kg/yr), which comparable in magnitude to upstream surface water loading. Advective loading for both pathways is greatest between RM 8 and 8.9 (Figure 10.2-39).

Cross-media comparison and statistical assessment of surface sediments, sediment traps, and suspended solids in surface water (Table 10.2-2a-b and Figure 10.2-37b) show that the concentrations in all media are statistically different.

10.2.9.4 Human and Ecological Risks Assciated with ~~Chromium~~TBT

TBT was identified as posing unacceptable risk to fish and benthic invertebrates, though these risks are localized in historical shipyard areas.